Mercury Water Quality Conceptual Model
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I. BACKGROUND

The Suisun Marsh is located in southern Solano County, California, approximately 35 miles northeast of San Francisco (Figure 1). The Marsh is one of the largest contiguous brackish marshes remaining in the United States. It is an approximately 116,000 acre mosaic of seasonally managed wetlands, unmanaged tidal wetlands, bays and sloughs bordered by upland grasslands. The marsh provides important habitat for more than 221 avian species, 45 mammalian species, 16 reptilian and amphibian species, and over 40 fish species (DFG 1989; Meng and Moyle 1993). The Suisun Marsh is located within the Bay-Delta estuary, consequently its water quality affects, and is affected by, California's two largest water supply systems, the federal Central Valley Project and the State Water Project, as well as other upstream diversions.

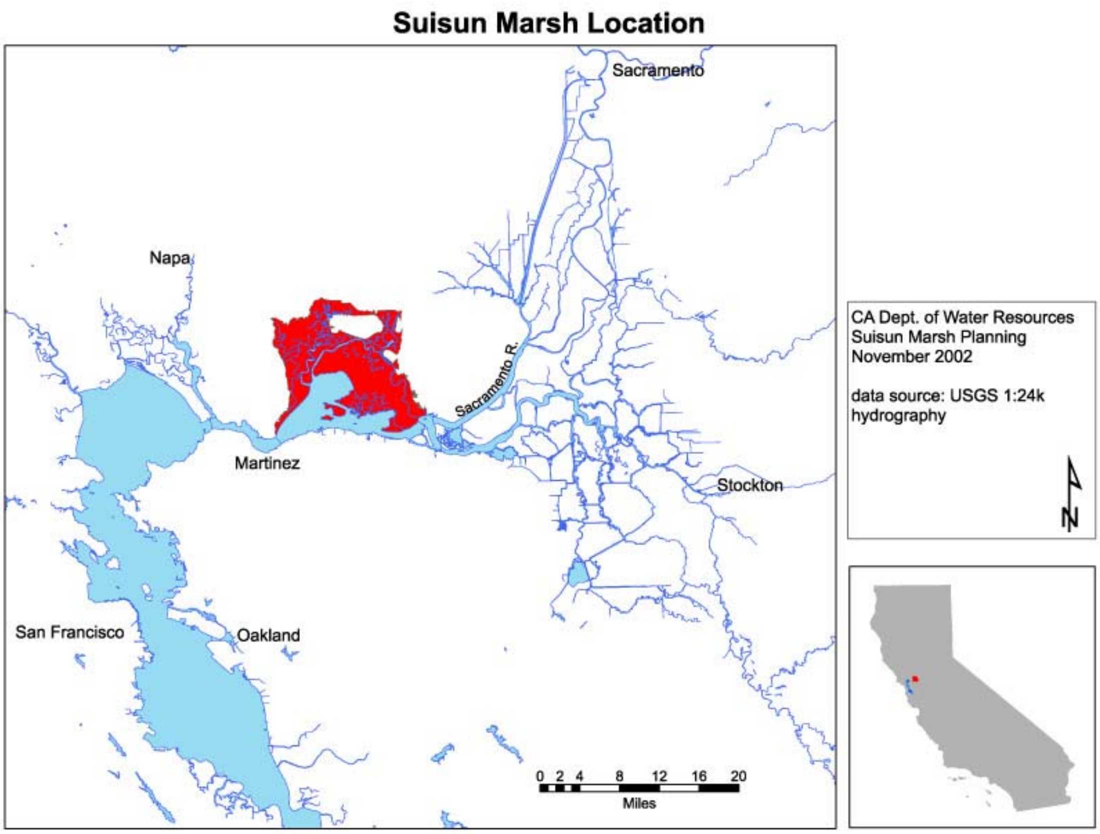

Figure 1. Suisun Marsh Location Map. Source: Department of Water Resources Suisun Marsh Program website. Available at: http://iep.water.ca.gov/suisun/map/index.html

Historically, Suisun Marsh contained more than 65,000 acres of tidal marsh (Goals Project 1999). Today the majority of the Suisun Marsh, approximately 52,000 acres, exists as seasonal wetlands in private ownership. Only 6,300 acres of Suisun Marsh remains as unmanaged tidal marsh. These tidal areas consist of fringing tidal marsh on the outboard side of levees and public lands owned by California Department of Fish and Game, the Solano County Open Space Foundation, and the U.S. Government. The primary goal of most seasonal wetland management is to provide habitat for waterfowl. Traditional waterfowl management systems in Suisun Marsh are based on flood-drain schedules to encourage the growth of waterfowl food plants. Techniques developed in the 1950s and 1960s are followed to decrease soil salinity and increase production of alkali bulrush, fathen, and brass buttons. Alkali bulrush is encouraged because it's tolerant to salinity and is considered to be a good duck food. Managers conduct leach cycles from the end of duck season until April or May to remove salts from the soils. During summer, pond bottoms remain dry to allow restrict growth of cattails and tules and to allow heavy equipment to disk pond bottoms. The length of the leaching cycles is used to control the composition of vegetation. A more detailed discussion of this process is described in the Managed Wetland Conceptual Model.

Competing resource needs between tidal and managed wetlands and water quality have made the Suisun Marsh one of the most highly regulated wildlife habitat areas in California, and has given it a prominent place in the CALFED Bay-Delta Program, a joint State-federal program formed to balance competing water needs and protect the Bay-Delta ecosystem. The Suisun Charter Group, formed at the request of CALFED, is in the process of preparing the *Habitat Management, Preservation and Restoration Plan for the Suisun Marsh* that would balance the needs of competing resources. The Charter has identified the following fundamental needs to be addressed in development of the Plan: protect and enhance 1) Pacific Flyway and existing wildlife values in managed wetlands, 2) endangered species recovery, and 3) water-project supply quality.

In order to ensure integration of science throughout the process, conceptual models have been prepared that describe our understanding of key habitats, species, or attributes found in Suisun Marsh. These conceptual models focus on current conditions of managed wetlands, tidal marsh, and four aspects of water quality (dissolved oxygen, organic carbon, mercury) and the fundamental physical processes driving water transport. The purpose of each of the conceptual models is to describe the current understanding (both what we do and don't know) of the processes and functions of the system/habitat. These models will provide a tool to evaluate effects of proposed Charter alternatives by helping to forecast the effects of future actions on forcing functions, rates, and applicability of processes, and system responses to those changes.

II. MERCURY WATER QUALITY MODEL - INTRODUCTION

This model describes the fundamental process of mercury cycling in the Suisun Marsh. It is intended to support the Suisun Marsh Charter planning process. The purposes of this model are to describe the current understanding of the processes of mercury cycling in Suisun Marsh and to provide a tool which can help evaluate the effects of potential Charter alternatives.

A. Model Organization

This paper is organized in the following manner:

I. Introduction
II. Linkages with other conceptual models
III. Conceptual Model of Current conditions: including a discussion of processes, functions, responses, uncertainties and assumptions
IV. Conclusions
V. Literature Cited

B. Mercury human health and wildlife concerns

Mercury exposure poses health risks for both humans and wildlife. There are three forms of mercury: elemental, inorganic, and organic compounds, each having different toxicological characteristics (Goyer 1991). Methyl mercury is the most important form of mercury in terms of toxicity and ability to biomagnify. Methyl mercury concentrations increase with each step in the food chain, whereas inorganic mercury is not readily transferred between trophic levels (Weiner et al 2003). Humans are exposed primarily through consumption of contaminated fish (Cooke et al 2004, Heim et al 2003, Johnson and Looker 2003). Concentrations of mercury found in the San Francisco Estuary are high enough to warrant concern for the health of humans and wildlife. The Office of Environmental Health Hazard Assessment (2004) has posted an interim advisory limiting consumption of fish from the San Francisco Bay and Delta region due to mercury contamination.

Mercury is a neurotoxicant, posing the greatest risk to developing embryos (Cooke et al 2004, Goyer 1991). All forms of mercury cross the placenta to the fetus; however, methyl mercury levels in fetal red blood cells are 30 percent higher than in maternal red blood cells. Exposure to mercury in *utero* or postnatal can cause irreversible neurotoxicity resulting in delayed motor skills, seizures, and other mental symptoms (Goyer 1991). In adults, the major health effects are neurotoxic and include numbness and tingling in the extremities, inability to walk, difficulty in swallowing and talking, weakness and fatigue, vision and hearing loss, tremors, and finally coma and death (Cooke et al 2004, Goyer 1991).

Wildlife species can be exposed to mercury through water, sediments or food sources. Aquatic habitats tend to be greater sources of mercury exposure than terrestrial habitats, due primarily to bioconcentration and bioaccumulation pathways (Davis et al. 2004). In aquatic systems, low trophic level species such as phytoplankton bioconcentrate mercury by accumulating mercury directly from the water (Cooke et al 2004, Davis et al, 2004). Higher trophic level species, such as piscivorous fish and birds, bioaccumulate mercury by ingesting mercury contaminated food sources. These species are the most at risk for mercury toxicity. Methyl mercury exposure has been found to cause reproductive impairment in many bird species (Cooke et al 2004, Davis et al 2004). Other wildlife species exhibit adverse effects of methyl mercury exposure including impaired learning, reduced social behavior, and impaired physical abilities such as difficulty flying, walking or standing (Cooke et al 2004, Davis et al 2004).

Due to the toxicity of mercury to both humans and wildlife, it is necessary to describe the current processes and forcing functions governing mercury cycling in the marsh. A description of these processes and functions will assist in evaluating how Charter alternatives may alter the current conditions of mercury cycling in the Marsh.

III. CONCEPTUAL MODEL

A. External Sources of Mercury

Mercury can enter the Suisun Marsh from four primary pathways: the Sacramento-San Joaquin Delta (Delta), coastal marine embayments, local watershed runoff, and the atmosphere (Figures 2a-2e).

The Delta

Mercury enters the Delta in the form of contaminated sediment deposits and contaminated runoff from the Coast Range and Sierra Nevada (Davis et al 2004, Heim et al 2003, Slotten at al 2002, Weiner et al 2003). The origin of the mercury contamination stems from the historic mining of mercury in the Coast Range and the subsequent use of elemental mercury for gold and silver extraction in the Sierra Nevada (Heim et al 2003, Marvin-DiPasquale and Agee 2003, Slotten et al 2002, Weiner et al 2003). Between 1846 and 1981 approximately 103.6 million kg of mercury were produced in California (Davis et al 2003). Losses to the environment from elemental mercury mining are estimated to range from about 10 to 30% (Weiner et al 2003) with an average of about 25% (Davis et al 2003) Average mercury losses may have been on the order of 34 million kg (Davis et al 2003). Elemental mercury was used in gold mining to trap and amalgamate gold. It is estimated that during the 1800's hydraulic mining and lode gold mining released about 3.6 to 6.0 million kg of mercury (Davis et al 2003). About 400 million m^3 of hydraulic mining debris deposited in northern San Francisco Bay, and an estimated half of this material still remains in the Bay (Davis et al 2003). Recent studies have determined that about 350-750 kg of mercury is still being transported annually in to the Bay-Delta from both the Coast Range and the Sierra Nevada.

Foe (2003) estimated the magnitude of the loads of mercury entering the Bay-Delta estuary from the Central Valley and exiting to San Francisco Bay and Southern California. The results were combined with the amount of mercury fluxing to and from sediment to calculate a methyl and total mercury budget for the Delta. Mercury exports to San Francisco Bay were based on samples collected at X2 (the location in the estuary with average bottom salinity of 2 psu), which was located in the shipping channel off Suisun Bay during much of the study. Foe determined that mercury export to Suisun is about 5 g/day methyl mercury and about 1,050 g/day total mercury.

Coastal Marine Embayments

Mercury contaminated sediment and water likely enters the Suisun Marsh from San Francisco Bay through tidal transport processes (WQ transport model, Enright 2004). San Pablo Bay is a 282-km^2 embayment with fringing wetlands located between Suisun Marsh and San Francisco Bay. Between 1856 and the late 1887 more than 250 million cubic meters of sediment from hydraulic gold mining was deposited in San Pablo Bay (Jaffe et al 2001). A recent study conducted by Marvin-DiPasquale et al (2003) used two approaches to distinguish the presence of hydraulic mining waste in San Pablo Bay, total Mercury concentrations and neodymium isotopic signature. All sites and depths had similar total Mercury concentrations (0.3–0.6 ppm) and geochemical signatures of mining debris. A bathymetric study of San Pablo Bay indicates that from 1951 through 1993 San Pablo bay was erosional as a result of decreased hydraulic mining debris and diminished sediment supply to the Delta and Bay (Jaffe et al 2001). Erosion and transport of these sediments may have provided a source of mercury to Suisun. Current sedimentation processes in San Pablo Bay are unknown, however, erosion of contaminated sediments may continue to provide a mercury source to Suisun Bay.

Watershed Runoff and Atmospheric Deposition

Other sources of mercury to the Marsh may include urban runoff, local watershed runoff, atmospheric deposition, and municipal and industrial effluents. The importance of these sources to the mercury loading of the Marsh is unknown, but may be significant. Crude estimates for stormwater loading from small tributaries to San Francisco Bay is 200 to 400 kg/yr (Davis et al 2003). Atmospheric deposition of mercury to the Marsh was calculated to be about 1.6 kg/yr. Atmospheric deposition is the sum of wet and dry deposition falling on water surfaces and indirect deposition on the watershed with subsequent runoff during storms (Cooke et al 2004). The following equation from Cooke et al (2004) was used to calculate the annual direct deposition rate for mercury on the Marsh:

$$Dt = (CwPyA)(1+Kd)$$

Dt = Total annual mercury deposition to Suisun Marsh (kg/yr)
Cw = concentration of mercury in precipitation (8.0 ng/L)
Py = Annual precipitation in Suisun Marsh (0.057 m/yr)
A = Surface area of Suisun Marsh ($3.6 \times 10^8\ m^2$)
Kd = Dry deposition coefficient (ratio of dry to wet deposition; assumed to be 1)

The direct wet atmospheric loads were calculated using a mercury concentration of 8.0 ng/L. This is the volume-weighted average mercury concentration in precipitation for 59 samples collected in the Bay Area between September 1999 and August 2000 (Tsai and Hoenicke 2001). The annual precipitation value for the marsh was based on the average annual precipitation for the period of record from December 4, 1950 to March 31, 2005 as calculated by the Western Regional Climate Center (WRCC 2005). The surface area of Suisun Marsh includes the surface area of the bays and sloughs, tidal wetlands, and managed wetlands. Dry deposition was assumed to be equal to wet deposition, as was done in Cooke et al (2004). Foe (2003) estimated the rate of atmospheric mercury deposition in the Delta to be slightly higher, about 2.8 kg/yr during wet years.

B. Existing Mercury Sources in Suisun Marsh

Suisun Bay, like the Delta, received massive amounts of mercury contaminated sediments from gold mining activities. Bathymetric surveys revealed that between 1867 and 1887, there was a net deposition of about 60 million cubic meters of sediment in the Suisun Bay area (Cappiella et al 2001). Most of this debris was from hydraulic gold mining in the Sierra Nevada, and is likely contaminated with mercury. Following the cessation of hydraulic mining and the construction of the water projects, sediment input to Suisun Bay decreased, and from 1887 to 1990 Suisun Bay was erosional. During this period, Suisun Bay lost more than 100 million cubic meters of sediment. Current sedimentation patterns in Suisun Marsh are unknown.

In 1999, Slotten et al (2000) sampled surficial sediments (top 1 cm) throughout Suisun Marsh and the Delta and analyzed the samples for total mercury. Mercury concentrations in the Marsh generally ranged from 0.20 to 0.33 ppm (dry wt), with one sample along Montezuma Slough containing a mercury concentration of 0.02 ppm (dry wt). In comparison, mercury concentrations in sediments in the Delta ranged from 0.15 to 0.20. Similar mercury concentrations were found by Heim et al (2003) in a 1999/2000 study. Sediment samples were collected from Suisun and Grizzly bays, and were found to have total mercury concentrations averaging 0.3 ppm (dry wt) with some sites above 0.5 ppm (dry wt). Hornberger et al (1999) found that the mercury concentration in surficial sediment from Grizzly Bay was 0.3 ppm. However, the concentration increased to 0.95 ppm at a depth of 30 cm. The mercury enriched zone persisted to about 80 cm before declining to a background concentration of 0.05 to 0.08 ppm. The higher mercury concentrations in sediments 30-80 cm deep were attributed to hydraulic mining debris.

C. Mercury Cycling In Suisun Marsh

Forms of Mercury

The internal cycling of mercury and methyl mercury within the Suisun Marsh has not been well documented. Figure 2 is a conceptual model of mercury cycling in the Suisun Marsh, based on a synthesis of recent investigations of mercury transport and cycling within the Sacramento-San Joaquin Delta. The major forms of mercury in the San Francisco Bay-Delta watershed are elemental or metallic mercury (Hg^0), mercuric mercury (Hg^{2+}) in complexes with organic and inorganic ligands, cinnabar or mineral mercury (HgS_s) and monomethyl mercury (CH_3Hg^+) referred to in this document as methyl mercury (Figure 2c). These forms, and additional minor forms, are collectively referred to as total mercury. Particulate total mercury is the dominant phase in waters of the Bay-Delta estuary, and much of the filter-passing total mercury is associated with colloids (Gill et al 2003). Waterborne total mercury and methyl mercury seem to be strongly associated with organic matter in the estuary (Wiener et al 2003).

The predominant forms of mercury entering the Delta through runoff are in the forms of HgS_s and Hg^0 (Heim 2003, Davis et al 2003). Runoff from the gold mining regions of Sierra Nevada is in the form of Hg^0. Hg^0 has low solubility in water, and appears to be relatively non-reactive in water (Davis 2003). Hg^0 must be oxidized to Hg^{2+} before it can be converted to methyl mercury. HgS_s is the predominant form present in runoff from mercury mining regions in the Coast Range. HgS_s must be converted to dissolved Hg^{2+} or a dissolved Hg-sulfide complex before it can be converted to methyl mercury. This is a slow process, however, dissolved organic carbon increases the solubility of HgS_s (Davis 2003).

Mercury methylation/demethylation processes

As discussed earlier, methyl mercury is the most important form of mercury with respect to wildlife and human health concerns. Methyl mercury is produced through a process referred to as methylation, addition of a methyl group to Hg^{2+}. Methylation is performed primarily by sulfate-reducing bacteria, which are found at the zones of transition from anoxic to oxic conditions in the water column or sediment (Davis et al 2003, Wiener et al 2003). Mercury demethylation is carried out by a much more diverse group of bacteria, which includes aerobes, methanogens, sulfate reducers, and likely others (Marvin-DiPasquale and Agee 2003).

Forcing functions or Limiting Factors

Several factors control mercury-methylation and methyl mercury-degradation dynamics in sediments. One important factor is the total mercury concentration and the form of mercury. Some ecosystems with low total mercury concentrations in water and sediment, such as the Everglades, have high rates of methyl mercury production and bioaccumulation. Other ecosystems with high concentrations of total mercury may have low or moderate concentrations of methyl mercury and bioaccumulation (Davis et al 2003). Heim et al (2003) compared the concentrations of methyl mercury and total mercury in sediments from the California coastal range, which was contaminated with cinnabar, and sediments from the Sierra Range, which was contaminated with refined mercury (elemental). He found that although the coast range had significantly higher concentrations of total mercury, the methyl mercury concentrations from the two sites were equivalent, suggesting mercury in the coastal range is significantly less available for methylation than the mercury in the Sierra range.

The factors that control mercury-methylation/demethylation processes are also critical to the methyl mercury concentrations. These factors can be grouped into two main categories. The first includes factors that affect the activity, distribution, and community composition of the microbes involved in mercury transformations, such as temperature, pH, salinity, redox and the availability of suitable electron donors (e.g. acetate, lactate, methanol, H_2) and acceptors (e.g. O_2, Fe(III), Mn(IV), SO_4^{2-}, CO_2) (Marvin-DiPasquale and Agee 2003). The second group of factors includes those that affect the availability of the substrate (mercury(II) or methyl mercury) to the methylating or demethylating bacteria, respectively (Marvin-DiPasquale and Agee 2003). These factors include mercury-species complexation with dissolved ligands (e.g. organics, polysulfides), binding or adsorption to solid organic and mineral phases, and pore water chemistry (e.g. pH, chloride, or sulfide concentration) that influences speciation (Marvin-DiPasquale and Agee 2003). Table 1 lists several of the factors and their relationship with methyl mercury production. However, the interaction of these factors and the results on methyl mercury production is not well understood.

Table 1. Factors that may influence mercury methylation/demethylation processes in Bay-Delta tidal wetlands

Variable	*Relationship with methylmercury production*	*References*
Oxygen	Sulfate reducing bacteria are the primary methylators and require anaerobic conditions	Compeau and Bartha 1985; Regnell and others 1996; Gilmour and others 1998; Choi and Bartha 1994
pH	Low pH associated with increased methylation	Xun and others 1987; Westcott and Kalff 1996; Rose and others 1999
Sulfate	In low sulfate waters, increased sulfate increases methylation.	Chen and others 1997
Sulfate	In high sulfate waters (e.g., estuaries) increased sulfate increases demethylation. Increased sulfide decreases methylation.	Oremland and others 1995; Benoit and others 1998
Dissolved organic carbon	High DOC in the water column may indicate high organic loading leading to high bacterial activity and anoxic sediments.	Watras and others 1995; Krabbenhoft and others 1995; Driscoll and others 1995
Dissolved organic carbon	Complexation of Hg species by DOC may increase dissolved concentrations without appreciably increasing biological uptake.	Barkay and others 1997
Temperature	In general, higher temperatures (up to 35 to 40 °C) result in higher bacterial activity.	Choi and Bartha 1994
Salinity	Bacterial and algal mercury uptake related to concentration of neutral species ($HgCl_2$, MeHgCl)	Barkay and others 1997; Mason and others 1995

Source: Davis et al 2003

Recently, studies have been conducted to increase our understanding of the complex methyl mercury production and degradation dynamics in the San Francisco Estuary. Results of one study indicated that mercury methylation in San Francisco Estuary sediments was most directly mediated by sulfur biogeochemistry, gross potential methyl mercury degradation rates, and possibly sediment pH, whereas methyl mercury degradation was most directly influenced by seasonal temperature variations (Marvin-DiPasquale and Agee 2003). A strong seasonal trend was seen for both mercury methylation and degradation processes. Methyl mercury production potential was greatest during winter, and decreased during spring and fall. This trend was related to both an increase in methyl mercury degradation, driven by increasing temperature, and to a build-up in pore water sulfide and solid phase reduced sulfur driven by increased sulfate reduction during the warmer seasons (Marvin-DiPasquale and Agee 2003). In a study in the central Delta, Heim et al (2003) found methyl mercury peaks in the sediment during summer and winter. The summer peak was larger, but short in duration (1-2 months), while the winter peak was lesser in magnitude, but longer in duration (3-4 months). Heim hypothesized that the seasonal changes in methyl mercury concentration were the result of fluctuations in microbial activity within the sediments.

Intertidal vegetated wetlands have been found to have significantly greater potential to methylate mercury than adjacent channels, mudflats, or open water. In a study in San Pablo bay, sediments from an intertidal marsh had a methylation/demethylation ratio more than 25 times that of all open-water locations (Marvin-DiPasquale et al 2003). In the Delta, Slotten et al (2002) found that flooded tracts characterized by dense submergent and/or emergent aquatic vegetation and highly organic sediments, had dramatically greater sediment methyl mercury than adjacent non-wetland control sites. These sites included all of the most elevated sediment methyl mercury samples, with vegetated wetlands tracts exhibiting up to ten times greater methyl mercury concentrations than adjacent control sediments (Slotten et al 2002). In Suisun, sediment samples were collected from the Ryer Island tidal marsh and the adjacent Grizzly Bay. Methyl mercury concentrations on Ryer Island were 2.15 ng/g (dry wt.) as compared to 0.30 ng/g (dry wt.) in the adjacent channel.

There are several characteristics of tidal wetlands that make them conducive to methyl mercury production (Figure 2e). There is a very large and almost continuous organically rich oxic-anoxic interface (where mercury methylation occurs) in the sediments of a tidal wetland. Smaller interior, or first-order, channels generally have limited water circulation leading to increased water temperature and residence time. Coupled with the large amounts of organic material this can lead to hypoxic water conditions. In the presence of reactive mercury these conditions are ideal for mercury methylation (Kelly et al. 1997). Field and lab based studies have suggested that organic carbon is positively correlated with methyl mercury in sediments (Heim et al 2003). Heim et al (2003) sampled three wetland areas in the Delta. Methyl mercury concentrations at the interior of all wetland areas studied were higher than concentrations at the exterior of the wetlands. In addition, the methyl mercury to total mercury ratio was highest at the interior of all three wetlands studied (Heim et al. 2003).

While tidal wetland areas in Suisun Marsh and the Delta have been shown to be high producers of methyl mercury, production of methyl mercury in the managed seasonal marshes has not been well documented (Figure 2d). As discussed in detail in the Managed Wetland Conceptual Model, the hydrology in the managed seasonal wetlands is very different than in adjacent tidal marshes. Managed seasonal wetlands are surrounded by levees and water is delivered through tide gates and along artificial channels. Water management usually consists of periods of prolonged flooding and periods of complete drying. While these managed wetlands may provide habitat for some tidal marsh species, their biogeochemical processes are likely very different from natural tidal marsh. The difference in hydroperiod between managed and tidal marsh likely exerts a significant influence on mercury methylation potential of the systems. Studies have found that newly flooded and reflooded terrestrial soils produce a surge in mercury methylation (Slotten et al. 2002). Snodgrass et al. (2000) found highest mercury levels in fish from shallow wetlands (30 cm) with relatively large water fluctuations. The study suggested that intermittent or periodic flooding associated with changes in water levels in wetlands can enhance methyl mercury production and/or bioavailability (Snodgrass et al. 2000). In a study of Coastal and Sierra range lakes, Heim et al (2003) found that permanent lakes were significantly lower in methyl mercury concentration than the seasonal lakes, in which some of the highest concentrations of methyl mercury occurred. Heim et al (2003) hypothesized that the increase in methyl mercury at newly flood areas is linked to increased microbial activity in response to a change in environmental conditions. Three changes in environmental conditions known to stimulate mercury methylation are 1) sudden death of vegetation supplying a large amount of organic carbon to become available for decomposition, 2) high decomposition leading to an increase in anaerobic habitat, and 3) mercury methylation stimulated by increased temperature (Kelly et al 1997). Snodgrass suggested that increases in methyl mercury are due to release of bound mercury from sediments during dry periods and uptake by biota when sediments are reflooded.

Alteration of the natural hydroperiod in managed systems also leads to changes in other variables that influence mercury methylation in wetlands. Drying of wetland soils often leads to accelerated decomposition of marsh litter, subsidence, oxidation of soils, and drastically lowered pH (Heitmeyer et al 1989), conditions known to stimulate mercury methylation (Davis et al 2003, Kelley et al 1997). The degree to which water management in seasonal wetlands affects the rates and processes of mercury methylation is unknown and is a critical link to understanding the mercury fluxes in Suisun Marsh.

D. Uncertainties/Questions

- Is Suisun bay erosional or depositional now? Are existing mercury deposits in Suisun being buried or eroded?
- Is the methyl mercury that is produced in the Marsh a source to the estuary or is the estuary a source to the Marsh?
- Within the Marsh, where will the exposure to biota be the highest? Managed wetlands, marshes, channels? Which species are most at risk?
- If tidal wetlands are created how can the methyl mercury exposure to biota be minimized? How can export to surrounding marshes and/or sloughs be minimized?
- Do the discharges from the managed wetlands that have low dissolved oxygen readings also have high methyl mercury concentrations and can the discharges be regulated to minimize the methyl mercury concentrations?
- Are there habitats in Suisun which are better mercury methylators? Can we learn something from these that will be useful in tidal marsh restoration?
- Do biota respond to periodic pulses of available methyl mercury or is it the long-term annual concentration that is critical?

E. Recognized Information Gaps

- The distribution and forms of mercury within Suisun Marsh have not been documented.
- The mercury transport mechanisms in the Marsh are not well understood.
- The mass balance of mercury and methyl mercury in the Marsh has not been determined.
- The relative contribution of methyl mercury production in managed wetlands and tidal wetlands has not been determined.

F. Suggested Targeted Research/Monitoring

- A network of water quality stations should be established to describe the methyl mercury distribution and pattern within the Marsh. Studies should be coupled with flow and turbidity measurements.
- Monitoring of methyl mercury concentrations in fish in the Marsh. Monitoring should include both short and long lived fish. Fish should be monitored once per year at a minimum.

IV. KEY FACTORS

Wetlands are known to be areas of high methyl mercury production (Heim et al 2003, Davis et al 2003, Weiner et al 2003, Marvin De-Pasquale et al 2003). The factors that influence methyl mercury production are numerous and not well understood. However, there are three key factors that appear to be critical to net methyl mercury production. These factors include total mercury concentration, speciation of the mercury, and level of activity of methylating bacteria. The level of activity of methylating bacteria is controlled by several other factors, but is generally greatest at the oxic-anoxic interface in the sediments. Hence, the area of the oxic-anoxic sediment interface in a given wetland should also be considered a primary factor in methyl mercury production.

V. KEY SCREENING QUESTIONS

Based on the key factors discussed above, the following key screening questions should be considered in screening SMIP alternatives:

- Are existing total mercury concentrations known for the given location? Is mercury speciation known?
- Will implementation of the alternative result in a change in the amount of oxic-anoxic interface in the sediments?
- Is implementation of the alternative likely to affect the level of activity of methylating bacteria (see Table 1)?

VI. CONCLUSIONS

Mercury contamination in the San Francisco Bay and Delta is clearly an issue of concern for both humans and wildlife. Research conducted to date has provided some information on the concentrations of mercury in the sediments of the Delta and Suisun; as well as an increased understanding of the processes involved in mercury cycling in tidal wetlands. However, research is needed to understand the mercury cycling processes occurring in managed seasonal wetlands and the associated forcing functions. It is critical that actions conducted as part of the Charter process incorporate monitoring to address the uncertainties/data gaps identified in this Conceptual Model.

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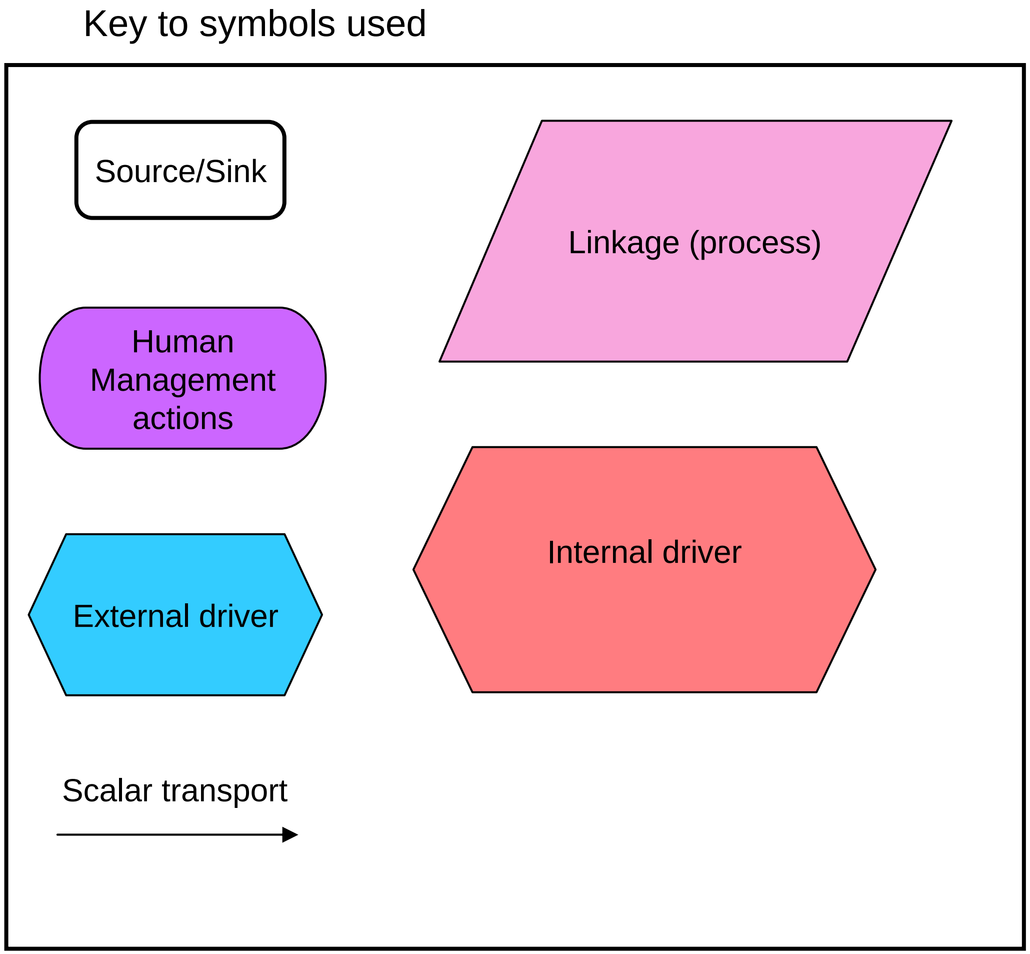

Figure 2a. Conceptual Model for Mercury in the Suisun Marsh

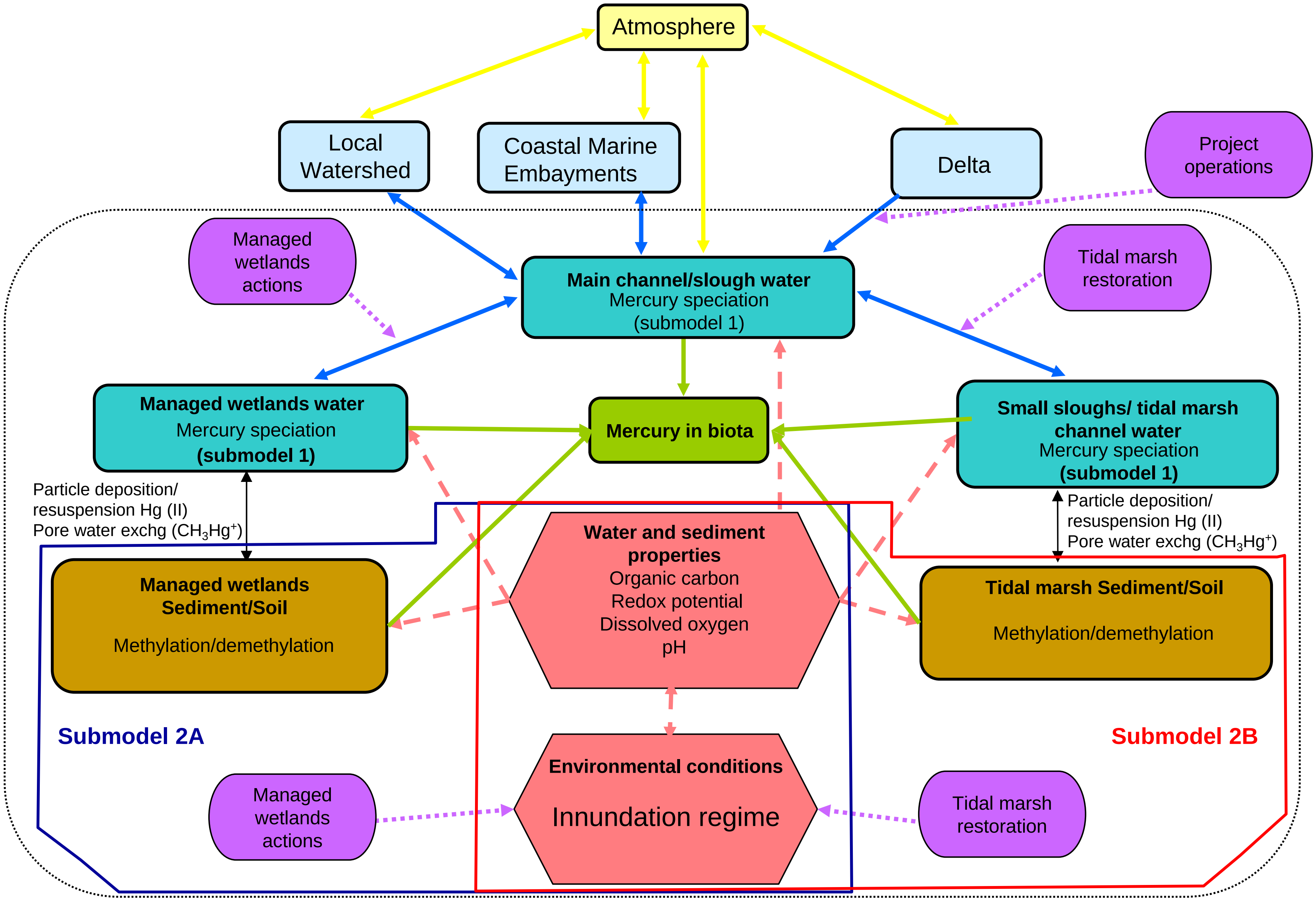

Figure 2b. Mercury transport in Suisun Marsh

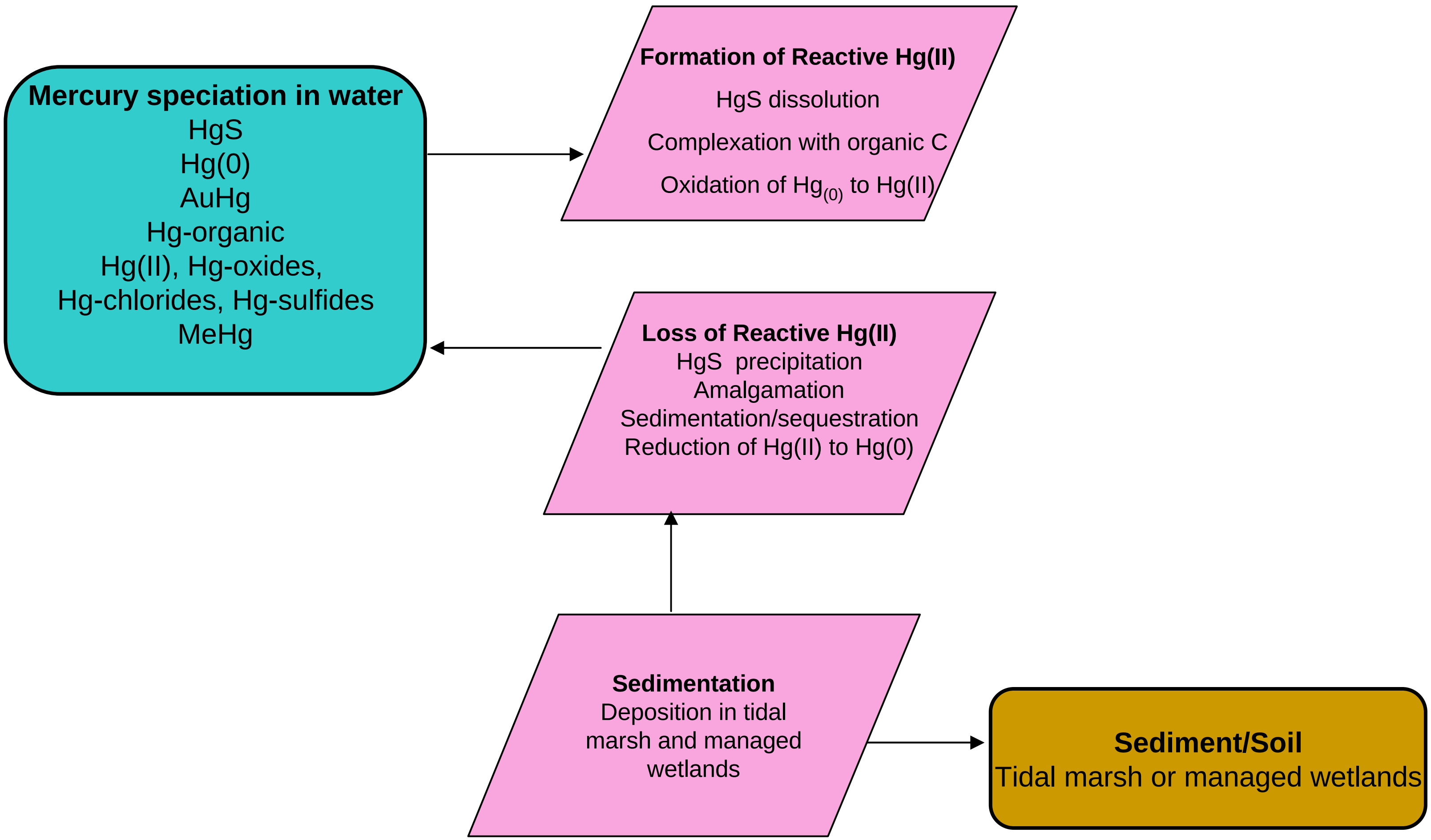

Figure 2c. Submodel 1 - Mercury speciation in the Suisun Marsh

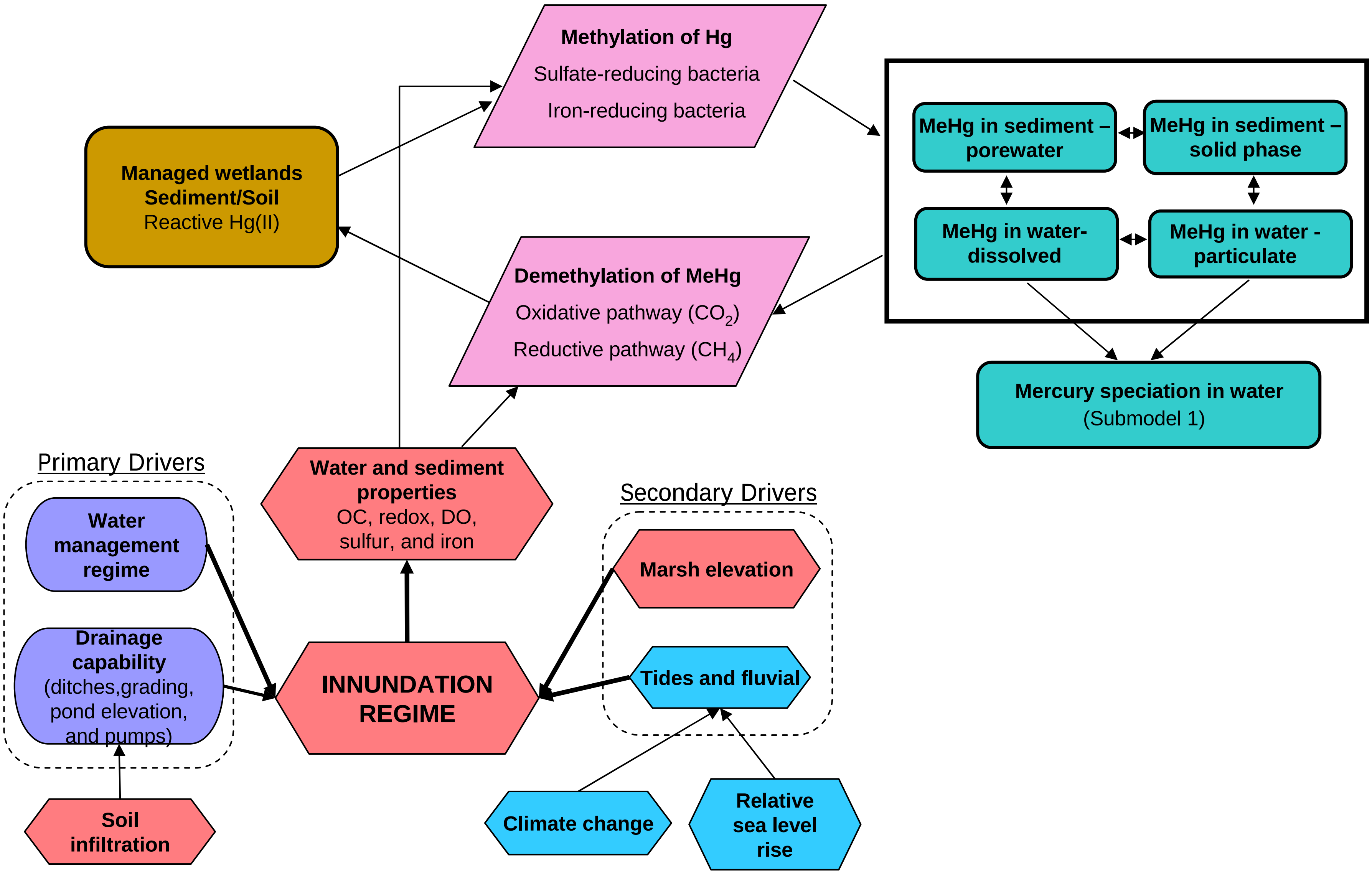

Figure 2d. Submodel 2A - Managed marsh inundation regime and mercury methylation processes

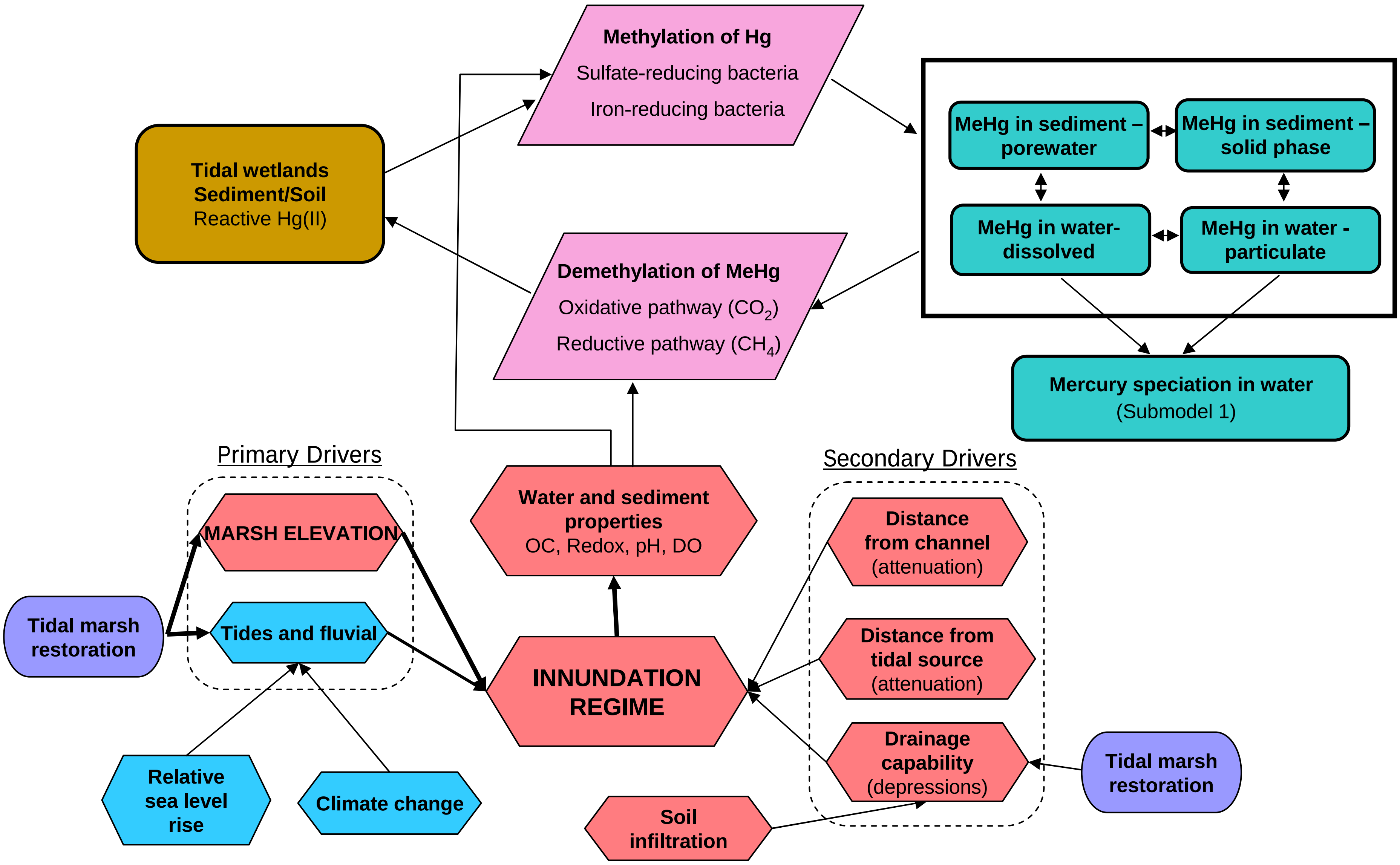

Figure 2e. Submodel 2B - Tidal marsh inundation regime and mercury methylation processes